

these complexes varies according to the metal and also with the pH. Thus the toxicity of cyanides and effluents containing cyanides (which commonly contain substantial amounts of heavy metal) is, as with ammonia, greatly influenced by pH, but is less well understood. It is clear, however (see Chapter 1), that the contribution of cyanides to the overall toxicity of complex industrial effluents can be identified, and is often significant.

Phenolic substances include the monohydric phenols (phenol, cresols and xylenols) and the dihydric phenols including catechols and resorcinols. They are found in a wide range of industrial effluents, and are particularly associated with gas and coke production, the refining of petroleum, power generation, many branches of the chemical industry and the production of glass, rubber, textiles and plastics. Alabaster and Lloyd (1980) and Buikema *et al.* (1979) provide detailed reviews of the literature. The latter review includes discussion of phenol derivatives which are used for specialised purposes such as pesticides. Apart from these, phenolic substances rarely occur as pollutants except as components of complex effluents which contain a variety of other polluting substances. In practice, therefore, the major concern is to determine the extent to which phenols contribute to the overall toxicity of an effluent, and an example was discussed in Chapter 1.

2.7.3 Pesticides

Pesticides are a diverse group of poisons of widely-varying chemical affinities, ranging from simple inorganic substances to complex organic molecules. Of the latter, some are natural metabolites, particularly of plants, while others are synthetic derivatives of natural products or completely synthetic substances produced in chemical factories under conditions which do not exist in the natural world. They have in common only that each pesticide is highly toxic to some forms of life and of intermediate or negligible toxicity to others, and that they have been widely introduced into the natural environment. Pesticides are introduced into aquatic systems by various means: incidentally in the course of their manufacture, and through discharge consequent upon their use. Surface water runoff from agricultural land and the side-effects of aerial spraying are especially important, and many serious pollution incidents arise through the accidental or negligent discharge of concentrated pesticide solutions which have been used for agricultural purposes such as sheep-dipping. Additionally, many pesticides are deliberately introduced into water bodies to kill undesirable organisms such as insect or molluscan vectors of human diseases, weeds, fish and algae.

Pesticides are also used in many industrial processes, for example in the manufacture of textiles and in the production and processing of perishable materials such as paper and timber products. They are therefore an important component of many industrial effluents. Because of their enormous diversity and their importance as pollutants, pesticides have attracted an enormous literature. Useful introductions are given, for example, by Khan (1977), Perring and Mellanby (1977), Brown (1978) and Muirhead-Thompson (1987). A valuable summary of pesticides as water

pollutants is given by Hellawell (1986). In many countries, the special significance of pesticides as pollutants (and as widely-used toxic chemicals in the working environment) has led to the development of strict controls on their use. One example is the Control of Pesticides Regulations, operated in Britain by the Ministry of Agriculture.

Insofar as it is possible to generalise about the polluting effects of such a diverse group of substances, the following points are perhaps of greatest significance. First, effective pesticides are more or less selective in their effects, that is they are extremely toxic to some forms of life and relatively harmless to others. Second, their modes of application vary according to the circumstances. In some cases, pesticides are applied in relatively high concentrations for relatively short periods. This pattern of application typically occurs where pesticides are applied to water to kill weeds, disease vectors or other undesirable organisms, or as an incidental effect of aerial spraying of crops. Here, the principal concern may be to determine their short-term toxicity to non-target organisms, and it may be advantageous to devise specific toxicity testing protocols in order to estimate the impact of the pesticides on the receiving-water biota (see Chapter 4). However, in lowland rivers draining agricultural areas, pesticides are more likely to be present at low but fairly consistent levels, and in this case the major areas of toxicological interest will be their potential sublethal effects, their capacity to accumulate in individual organisms and via the food chain, and the development of resistance through acclimation and/or genetic adaptation. Many pesticides are known to be refractory to chemical and biological degradation, and their persistence in the environment has for many years been a cause for concern. Probably the best example of this is the well-known case of DDT, which has been used in such enormous quantities in the last 50 years that no part of the world is now free from measurable contamination, and its manufacture and use in many countries are now banned or severely restricted. One consequence of this, however, is that other pesticides are used as substitutes; many of these are of much more recent discovery than DDT, and may eventually be found to be equally or more dangerous when sufficient information about them has been accumulated.

2.8 Suspended Solids

Virtually all effluents contain suspended particulate matter, but especially those associated with mining and quarrying for coal, china clay, stone and other mineral materials. Dredging, engineering works and boat traffic commonly introduce particulate matter into suspension. Storm-water drainage and surface water runoff also contribute substantial loads. Suspended matter may be organic or inert, and some forms are chemically reactive (for example, the ferric hydroxide precipitate associated with acid mine drainage (see Section 2.9)). The present discussion is confined to the physical effects of suspended solid matter. The effects of suspended matter on the receiving water biota are both direct and indirect. Direct effects include physical abrasion of body surfaces, and especially of delicate structures

such as gills. Physical damage of this kind interferes with respiration and renders the animals susceptible to infections. High levels of suspended particulates may interfere with the filter-feeding mechanisms of invertebrates, and possibly with the feeding of fish which locate their food visually. In laboratory experiments high levels of suspended solids can kill fish. The concentrations of suspended solids at which these effects occur vary with the species and the nature of the particulate matter. Herbert and Merkens (1961) found that kaolin and diatomaceous earth in suspension at a concentration of 270 mg l^{-1} caused substantial mortality to trout over a period of 10–15 days, but other investigations have shown large differences between the lethal concentrations of different types of suspended matter (Alabaster and Lloyd, 1980).

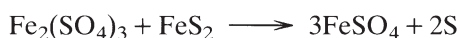
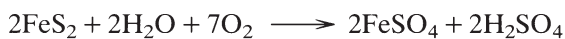
Indirect effects are mainly due to increased turbidity and the blanketing effect of the particulates when they eventually settle. Increased turbidity will reduce or prevent photosynthesis, leading to a reduction in primary productivity or the complete elimination of plants. Alternatively, certain forms of silting can, depending on the physical conditions, bring about major changes in the community by promoting the formation of stable weed beds. Salmonid fishes require aerated gravel beds for egg-laying sites, and the silting of gravel beds can eliminate salmonid populations by depriving the fish of suitable nest sites. In many rivers, salmonid population density is governed by the availability of nest sites rather than by biological or chemical factors. Invertebrate distribution patterns are profoundly influenced by the size of the particles composing the substratum (Hynes, 1970). Mayfly nymphs with exposed abdominal gills (e.g. *Ecdyonurus*, *Rhithrogena*, *Ephemerella*) may be replaced by species with covered gills or which are adapted for burrowing (e.g. *Caenis*, *Ephemera*). Insects which crawl upon the substratum may generally be disadvantaged in favour of species whose means of locomotion is better suited to a soft substratum, such as leeches, oligochaetes and some molluscs. Thus the input of even inert fine particulates can readily bring about major community changes. Herbert *et al.* (1961) described the effects of china clay wastes on the ecology of trout streams, and discussed the mechanisms by which alterations occurred. Other field and laboratory studies are reviewed by Alabaster (1972) and Alabaster and Lloyd (1980). The majority of unpolluted British rivers contain fewer than 50 mg suspended solids per litre of water, and about half have under 30 mg l^{-1} . Alabaster and Lloyd (1980) tentatively suggest that waters containing more than 80 mg l^{-1} are unlikely to support good fisheries.

2.9 Extreme pH and Acidification

Many effluents, especially if untreated, are strongly acidic or alkaline. All natural waters have some buffering capacity, that is the ability to absorb acid or alkaline inputs without undergoing a change in pH. This buffering capacity is usually expressed in terms of the acidity (ability to neutralise alkalis) and alkalinity (ability to neutralise acids) of the water, and is determined by titration in the presence of a suitable indicator. The relationship between the pH, acidity and alkalinity of a

water is not simple. Acid waters ($\text{pH} < 7$) can have measurable alkalinity, and alkaline waters ($\text{pH} > 7$) can have measurable acidity. Where the buffering capacity of the water is exceeded by the input of an effluent, the pH of the water will change. Unpolluted natural waters show a pH range from 3.0 to 11.0 or more; those lying between 5.0 and 9.0 generally support a diverse assemblage of species and this range may be considered broadly acceptable (Alabaster and Lloyd, 1980). This does not mean, however, that pH changes within the range 5.0–9.0 are of no consequence. For example, pH is an important determinant of the distribution patterns of aquatic species, as can be seen from the study of invertebrate communities in unpolluted rivers (Haines, 1981; Sutcliffe, 1983; Sutcliffe and Carrick, 1973). In addition, apparently small changes in pH can have major effects on the toxicity of pollutants such as ammonia, so that the effect of a given level of pollutant can vary, depending upon pH, from being scarcely noticeable to being extremely serious.

A very common form of pollution involving extreme pH is acid mine drainage. Coal mines are the most common source of acid mine drainage, but it can occur wherever mineral ores are mined. A series of reactions commencing with the oxidation of the common mineral pyrite (FeS_2) is responsible, and certain autotrophic bacteria are closely involved (Lundgren *et al.*, 1972). The reactions may be summarised:



These reactions occur chemically, but only very slowly at acid pH values. The sequence of reactions may therefore be self-limiting. However, certain bacteria, especially *Thiobacillus thiooxidans* and *Ferrobacillus ferrooxidans* possess enzymes which bring about these reactions rapidly. These bacteria are abundant in acid mine waters, especially since they are tolerant of low pH whereas many species commonly found in soil and water are not. Further, they are autotrophic, that is they require no substantial amount of organic matter for growth, and soon dominate the flora.

The effects of acid mine drainage are threefold. The low pH itself has adverse effects on the receiving water flora and fauna. It also promotes the solubilisation of heavy metals, which exert their own toxic effects. Third, as the drainage water is diluted and the pH rises, ferric hydroxide precipitates and discolours the water, producing the effects of suspended particles. As the hydroxide settles, it forms a gelatinous layer over and within the substratum, causing both direct and indirect

effects on the receiving water community. In general, the result is a marked reduction in species diversity and biomass in the affected areas (Koryak *et al.*, 1972; Letterman and Mitsch, 1978; Scullion and Edwards, 1980). The effect on the receiving water community is influenced by the alkalinity of the water or the presence of nearby alkaline discharges; where the acid is neutralised, the effects are due to the ferric hydroxide alone, rather than to the combined effect of hydroxide and low pH.

A phenomenon which has recently received widespread attention is that water bodies in some parts of the world, especially in North America and Northern Europe, appear to be becoming steadily more acidic. The reason, it is suggested, is that airborne pollutants from industrial areas are transported by prevailing winds to areas remote from their source, and precipitated in 'acid rain'. The subject has caused much controversy both within the scientific community and among the general public, particularly since the problem not only has grave implications for important economic interests, but has already given rise to differences between the governments of some countries. It is not yet clear, for example, whether the main cause is sulphur dioxide emissions from power stations, or motor vehicle exhausts. Some countries have already begun expensive programmes to limit SO₂ emissions, while others argue that the extent of acid rain is not controlled by the amount of SO₂ emitted but by chemical reactions in the atmosphere and that the reduction of SO₂ emissions would therefore not lead to a decrease in acid rain. It is also clear that certain forestry practices lead to acid runoff from the soil into surface waters, so that the source of the problem may lie not in distant industrial areas, but within the affected areas themselves. These controversies do not directly concern us, but it is appropriate to consider the extent to which natural waters are becoming acidified, whether or not this is a natural phenomenon, and what its biological consequences may be.

The consequences of acidification are undoubtedly serious. Haines (1981) and Howells (1990) provide general reviews, and Baker and Schofield (1985) discuss the impact of acidification on North American fish populations. It is generally accepted that as pH decreases, both the diversity of species and the overall productivity of aquatic ecosystems decline. These phenomena can readily be observed through contemporary studies on waters of different pH values. Several mechanisms appear to be involved. First, every species has its own zone of tolerance to pH. The pH range which permits survival may be wider than that which will permit successful reproduction. This is particularly well documented for fish, both in laboratory experiments and through field observations (Baker and Schofield, 1985). Sutcliffe (1983) draws attention to the important influence of pH on the mechanisms of ionic regulation in aquatic species. Naturally-acid waters tend to be found in areas of base-poor geology and tend to be low in nutrients, dissolved

ions and buffering capacity. Therefore they have low primary productivity, present unusual difficulty to animals and plants in maintaining ionic balance, and are particularly susceptible to extraneous inputs of acid. The distribution of invertebrates in relation to pH was investigated in a catchment in northern England (Sutcliffe, 1983; Sutcliffe and Carrick, 1973) and revealed a pattern which is widely repeated elsewhere (Haines, 1981). Phytoplankton, zooplankton and macrophytes behave similarly in relation to pH. Generally, the pattern is of reduced species diversity as pH declines, although in favourable circumstances acid-tolerant species may become locally abundant. Undoubtedly, an apparently small change in pH can bring about major changes in community structure. (It may be appropriate to recall that a reduction of 1 pH unit represents a tenfold increase in hydrogen ion concentration; and that a pH change from, say, 6.9 to 6.6 means that the hydrogen ion concentration has approximately doubled.)

In addition, low pH values in theory could strongly and progressively reduce the rate of decomposition of organic detritus, presumably through the effect of low pH on the fungal and bacterial organisms responsible for this process (Haines, 1981). Many aquatic ecosystems depend on the decomposition of allochthonous detritus (i.e. organic material from outside the system, such as fallen leaves) as the main source of energy for the animals in the system. Primary (photosynthetic) production of organic matter in acid waters is naturally low, being normally limited by the low availability of nutrients; it may also be itself inhibited by the susceptibility of phytoplankton and macrophyte species to low pH. The alternative source of energy for animals, organic detritus, is only available in the presence of microbial decomposers, since animals cannot digest plant material unaided. Therefore the overall productivity of the system will decline. However, Gilmour (1992), reviewing the results of experimental acidification studies, concluded that the effects of acidification on microbial decomposition processes do not in practice become significant until the pH drops below 5, by which time other more direct effects on the fauna and flora will already have taken place. Field studies on acidified lakes in Europe largely point towards a similar conclusion (Howells, 1990). Indeed, such evidence as exists suggests that microbial decomposers and their associated biogeochemical processes are more robust to reduced pH than are higher organisms, and that nutrient cycling may actually be accelerated by low pH.

A third effect of acidification is to increase the threat of heavy metal toxicity. Aluminium, a metal which does not commonly cause serious problems of toxicity to aquatic life, has received particular attention. Burrows (1977) and Odonell *et al.* (1984) have summarised the role of aluminium as a toxic water pollutant. Although it is a very abundant metal, forming up to 7% of the earth's crust, it is highly reactive and readily forms stable compounds of very low solubility. In most natural

waters which have been investigated, the dissolved aluminium concentration lies between 0.05 and 10 mg l⁻¹. Within the pH range of approximately 5.5 to 7.0, aluminium is practically non-toxic and is certainly harmless at the concentrations found in most waters. The chemistry of aluminium and its compounds at low pH is poorly understood, but it appears that as pH falls aluminium compounds become more soluble and that the proportion of free aluminium rises. Below pH 4, the toxic effects of free hydrogen ion are so severe that the presence of aluminium is probably of little significance. However, between pH 4 and 5.5, the toxicity of aluminium is high, reaching a maximum at around pH 5. At this pH, the level of aluminium naturally present in water is acutely toxic to fish. High levels of calcium appear to offer some protection, although acid waters with high calcium levels are unusual. Aluminium toxicity is likely, therefore, to be a major contributor to the effects of acidification. Surges of acidity in upland rivers associated with snowmelt are not uncommon, and can be particularly damaging to salmonid fish populations which tend to be breeding at the time when snowmelt is likely to occur, leaving young fry vulnerable (Howells, 1990).

More rarely, aluminium can also cause toxicity in alkaline waters, as the solubility of aluminium compounds also increases with pH above 7.0. Aluminium compounds are widely used in the treatment of potable waters, and discharges from water-treatment works can have adverse consequences (Hunter *et al.*, 1980).

Finally, the question of whether acidification is a recent phenomenon, and whether it is a cyclic or an irreversible change, can only be answered indirectly. Accurate measurement of pH, particularly in weakly-buffered solutions like natural surface waters, has only been possible for about 30 years, and even within this timespan few water bodies have been studied systematically. It is also difficult to distinguish long-term trends from diurnal or seasonal pH variations, so the data available even from recent measurements of surface water pH are not altogether reliable. However, contemporary observations on the distribution of diatoms in relation to water pH show that the diatom community shows distinct qualitative changes according to the pH of the water. Analysis of the diatom community therefore indicates approximately the pH of the water. The characteristic siliceous frustules of diatoms accumulate at the bottom of lakes and decay only very slowly, so analysis of a core of lake sediment allows the changes in the diatom community over long periods of time to be determined. Thus, indirectly, any changes in the pH of the water may be inferred. Flower and Battarbee (1983) used this technique to estimate changes in the pH of two lakes in Scotland over several hundred years (Figure 2.9), and their results are typical of many studies. It appears that the pH of lake water declines slowly for natural reasons; rain is naturally acidic. However, the rate of decline of lake pH

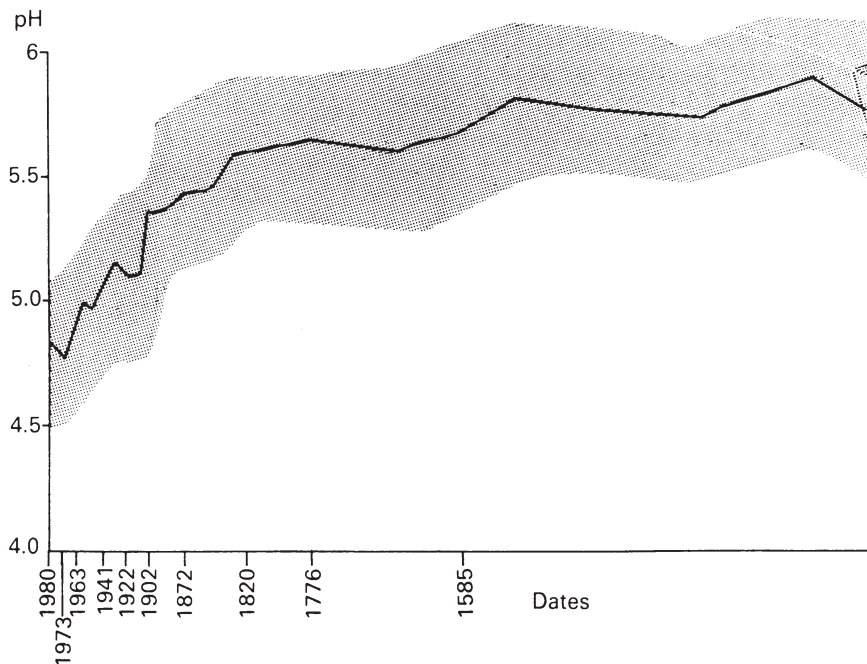


Figure 2.9 Changes in pH of a lake in Scotland with time, inferred from the composition of the diatom community in lake sediment cores at different depths (Flower and Battarbee, 1983). (Reproduced by permission from *Nature*, 305, pp. 130–132; copyright © 1983 Macmillan Magazines Ltd)

appears to have accelerated rapidly within the last 150 years or so, corresponding to the period during which Britain became heavily industrialised. Further discussion of many aspects of acidification is given by Howells (1990).

2.10 Detergents

Synthetic detergents are an interesting group of pollutants because they were virtually unknown before 1945, yet within a few years became responsible for some spectacular water pollution problems which, unusually, came rapidly to the attention of the general public. The alkylbenzene sulphonate detergents (Figure 2.10) rapidly replaced soap as domestic and industrial cleaning agents because of their cheapness and greater efficiency, and particularly because they did not cause precipitation of calcium salts in areas supplied with hard water. Unfortunately they were not readily broken down by sewage treatment processes, giving rise to problems of toxicity to the receiving-water biota, and of foaming in watercourses and treatment works.

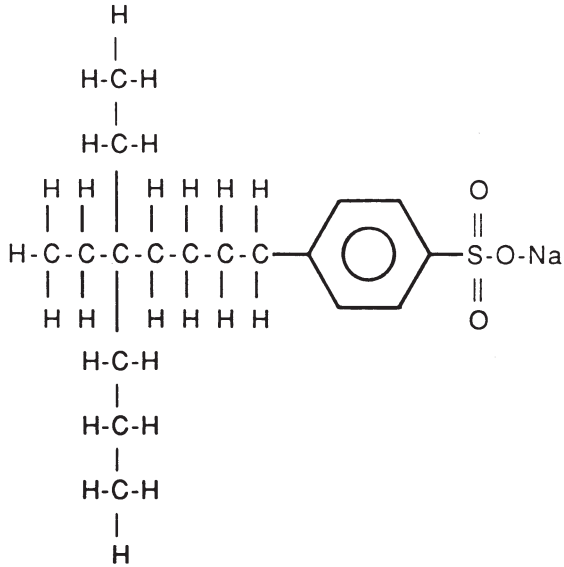
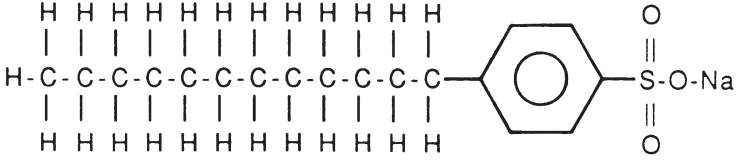


Figure 2.10 Molecular structure of (above) a typical 'soft' (biodegradable) linear alkylate sulphonate detergent and (below) a 'hard' alkylbenzene sulphonate detergent

In areas where industrial usage of detergents was pronounced (for example, in textile-processing industries) whole towns were frequently covered in detergent foam; in waste treatment works, a number of serious accidents occurred through, for example, operatives falling into sedimentation tanks which were concealed under a thick layer of foam. Consultation between regulatory authorities and the detergent manufacturers led to research which showed that modifications to the manufacturing process could produce *linear alkylate sulphonate* (LAS) detergents (Figure 2.10), which were rapidly degraded in conventional waste treatment plants. From 1965 onwards, 'soft' or biodegradable detergents were introduced for domestic use, and although these are generally more toxic to aquatic organisms, their unbranched hydrocarbon chain is more readily broken

down in treatment processes and in practice, toxicity and foaming problems had largely disappeared by the early 1970s. Synthetic detergents remain significant causes of pollution in some circumstances however. Their toxicity to aquatic organisms is reviewed by Abel (1974). ‘Soft’ detergents are not suitable for use in certain industrial processes. Detergents are widely used as components of oil-dispersants, particularly in coastal and marine habitats, and are often more toxic to aquatic organisms than the oil itself (see Chapter 7). Finally, some components of detergent formulations exert adverse effects of their own. The best known example is the high level of phosphate found in many formulations (see Section 2.4). Less widely appreciated are the adverse effects of boron, from perborate additives to detergent formulations, which can cause adverse effects on crops if contaminated surface waters are used for irrigation (Lester, 1975; see also Table 2.2).

2.11 Oil and Petroleum Products

Oil pollution is commonly perceived as being a problem associated mainly with the marine environment, but in fact oil and related substances account for about one-quarter of reported pollution incidents of fresh waters in Britain (NRA, 1994a); and probably the pattern is not much different in other parts of the world. Also, about one-quarter of all the oil released to the seas by human activity is estimated to enter via rivers. The sources of oil pollution are the usual ones—illegal, negligent or accidental discharges, plant failures and so on—but to these must be added inputs from roads and railways, particularly in the case of traffic accidents, and discharges associated with oil extraction from inland oilfields and processing plants.

The effects of oil discharges vary enormously, because the characteristics of the receiving waters and the various types of oil are themselves very variable. Fortunately many small discharges have relatively trivial effects, but substantial discharges can have severe impacts. Some case studies are reviewed by Green and Trett (1989). Generally, light oils are relatively volatile and disperse quickly, though they are extremely toxic and can cause severe local damage. Fast-flowing waters often recover rapidly, over a few weeks or months, but lentic waters are much more susceptible to long-term damage. Heavy oils are less toxic, but can have marked physical effects in the substratum and the banksides, which of course will be reflected in changes to the biological community. Heavy oils may remain *in situ* for long periods; they do, over time, decompose biologically and chemically, but in so doing impose a high BOD. Frequently, the effects of oil pollution are not dissimilar from those of heavy organic pollution, with varying levels of toxic effects adding to the overall impact. Oils, unlike most forms of water pollution, can also have damaging effects on terrestrial organisms, such as plants, birds and mammals living at the water’s edge, since heavy oil contamination usually leads to deposition of oil on the banks of lakes or rivers.

Water pollution biology

Generally, the use of oil dispersants, which are commonly used in dealing with marine spills, is avoided in freshwater because of their high toxicity. Also, the physical conditions of freshwater habitats are often amenable to containment and recovery of oil by booms, skimmers and/or the use of absorbent materials.

Biological Monitoring of Water Quality

3.1 The Conceptual Basis of Biological Monitoring

We have by now seen ample evidence that the levels of abundance and patterns of distribution of aquatic organisms may be affected by pollution of the water in which they live. Hellowell (1977, 1978) summarised diagrammatically the changes which may occur in a community subject to pollution (Figure 3.1). Which, if any, of these responses occur will depend upon the nature and severity of the pollution, and upon the relative susceptibility of the species within the community to specific kinds of environmental alteration. It follows that given suitable techniques of sampling and data analysis, monitoring of the biological characteristics of waters might indicate the occurrence of ecologically significant environmental changes—including the incidence of pollution—which may otherwise be undetected.

This basic idea has been understood and utilised in monitoring the effects of pollution since the early years of the twentieth century (Cairns and Pratt, 1993), although many biologists have felt that the techniques of biological monitoring have not, at least until quite recently, been accepted and developed as rapidly as they might have been. There are probably several reasons for this. First, chemists and physical scientists are frequently sceptical of biological results which can sometimes appear to them to lack precision, and frequently require expert interpretation. Biologists themselves, perhaps under pressure to produce results quickly and economically, frequently have in fact failed to follow the best practice and have on occasions produced ambiguous, misleading or inaccurate results. In a recent analysis of published studies, Resh and McElravy (1993) found a number of cases where, for various reasons, some aspects of the monitoring programmes reviewed were

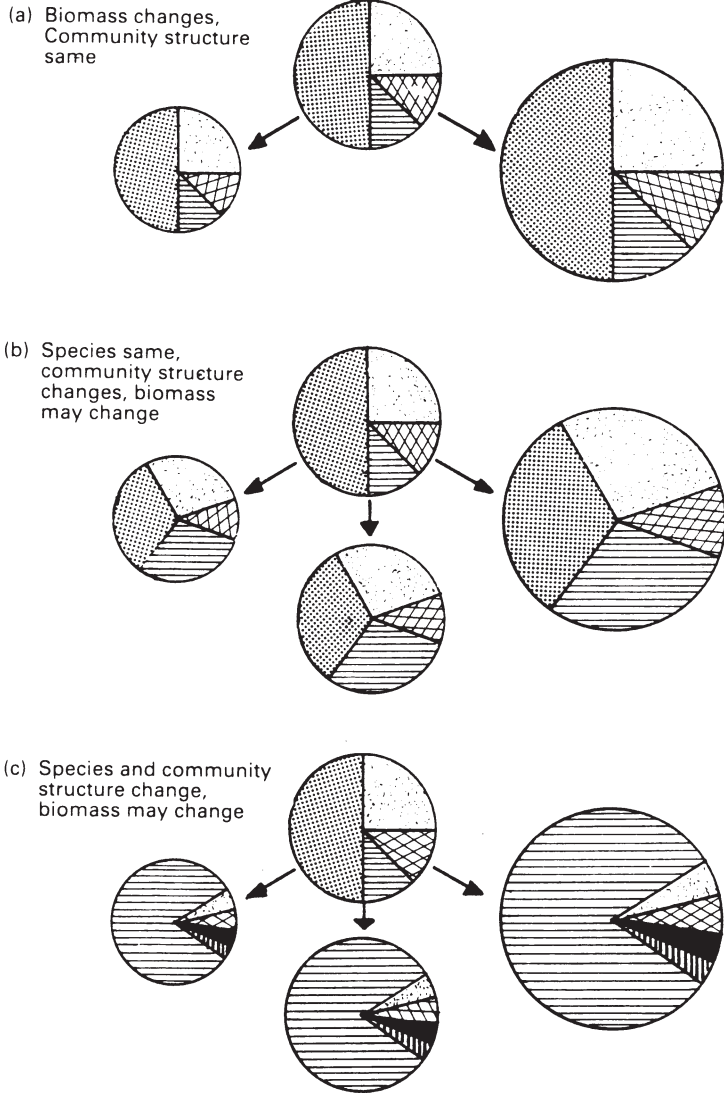


Figure 3.1 Possible responses of a community to environmental change (from Hellawell, 1978)

inadequate in one way or another, casting doubt on the validity of the results. Insofar as biological monitoring procedures have been adopted in the past by regulatory agencies, biologists have encountered resistance from legislators and administrators who are reluctant to modify regulatory practices, laws and regulations to take account of improvements in techniques, preferring instead to rely on 'established' methodology which more recent research shows can be greatly

improved upon. Finally, there is the stubborn belief that biological monitoring is prohibitively expensive—it certainly is demanding of trained human resources. However, in a very illuminating discussion of this and some of the other points raised above, Brinkhurst (1993) produces evidence to show that the cost-effectiveness of biological monitoring techniques actually compares favourably with physical and chemical methods.

Perhaps optimistically, Rosenberg and Resh (1993) express the view that 'Biomonitoring may have come of age with the recent adoption, by North American and European governments, of national programs of environmental monitoring and assessment that include the use of aquatic biota'. Certainly an impressive consensus has been developing over the last decade or so among actual practitioners of biological monitoring. In fact, this consensus has been in existence among specialists for much longer than that, though it has not always been appreciated by those who wish to utilise their services but who do not wish to pay the necessary price in patience, financial resources or flexibility of attitude. We should also remember that in any field in which new ideas and techniques are being constantly developed, the consensus must always lag a little behind the best approaches, and must itself evolve rather than be fixed in stone. The evolution of this consensus is perhaps best traced through the literature of biomonitoring during the last 20 years (see, for example, Hellawell (1977, 1986); Metcalfe-Smith, (1994); Rosenberg and Resh, (1993)). Now that techniques of biological monitoring are becoming more widely accepted, there are also specific recommendations published in accessible form by reputable agencies, which are a useful source of guidance to the relatively inexperienced. These include APHA (1995), ISO (1985, 1991) and DIN (1983a, b, 1987, 1990). The purpose of this chapter is to give a general introduction to the best current practice. Biological monitoring is a powerful tool, but a very dangerous one if not used correctly.

Biological monitoring programmes are carried out for a variety of reasons. For example, most regulatory agencies, such as water authorities, routinely monitor the biota of the waters for which they are responsible. Typically, sampling stations are surveyed at intervals of between one and six months. Information is gathered on the presence and relative abundance of species, and may be used to derive numerical values, for example of species diversity or biotic index (see Section 3.4), to facilitate spatial or temporal comparisons. Significant alterations from previously-established conditions may indicate the need for further investigation, and lead to action designed to preserve or improve the existing water quality. Sites known to be at special risk from existing pollution sources, and sites of special commercial or conservation value, for example salmon rivers, may receive special attention such as more frequent or more extensive surveillance. Proposed new developments, such as the siting of new effluent discharges, abstraction from or canalisation of watercourses, may be preceded by programmes of biological surveillance to establish 'baseline' biological conditions of the affected water. After implementation, the biota